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Quantifying residual forest structures following retention harvesting in northeast Minnesota using Landsat sensor data

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DEDICATION

To Bryce Gonzales.

"May your trails be crooked, winding,

lonesome, dangerous, leading to the

most amazing view. May your mountains

rise into and above the clouds.

-Edward Abbey

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CHAPTER I. GENERAL INTRODUCTION

Background

Forest simplification and Minnesota forest history

Intensive wood production has resulted in the simplification of forest structure and composition in forests worldwide (Puettmann et al. 2009). Disturbances that occur from industrial forestry can be drastically different from natural disturbance regimes (Lindenmayer and Franklin 2002), and often the industrial forestry approach is analogous to the conventional agricultural model where simplification is the goal (Smith et al. 1997). Traditional harvesting such as clear-cutting often removes much more of the ecosystem, does so more uniformly, and more frequently than natural disturbances (Franklin et al. 2000) These intensive management activities result in a lack of complexity in stands and across landscapes, which feeds back through ecosystem processes to carry a high risk of reducing key environmental services such as clean water, prevention of soil erosion, pest control, pollination, carbon storage, and local ecosystem resilience and stability (Thompson et al. 2011). In addition to a loss in ecosystem services, there is often less biodiversity in intensively managed forests. A comparison of abundance of insects, birds, mammals, fungi, plants and lichens between intensively managed forest in Sweden and Finland and adjacent natural forest in Russia showed substantially lower species numbers in managed forests, partially attributed to the homogenization and lack of structure in even-aged monocultures (Berg et al. 1994, Angelstam 1996).

The simplification of forest biophysical structure and composition is a global phenomenon (Puettmann et al. 2009). Forest simplification and homogenization within the

Great Lakes region began soon after European settlement (circa 1850; Cornett et al. 2001, Schulte et al. 2007), with extensive logging activities underway by the beginning of the 20th century; a period known as the cutover (Zon 1925). Logging ranged from selective harvesting of economically valuable species such as white and red pine (*Pinus strobus, Pinus resinosa*; Pastor et al. 2005) to heavy partial cuts and clear-cutting, which was often followed by repeated uncontrolled slash fires (Stearns 1997). These events have diminished the area of unharvested stands and stands in the old-growth stage of development in Minnesota to ~0.2% and 2% of their pre-European settlement extent, respectively (Frelich 1995). In addition, management of aspen for pulpwood has led to rotation ages of ± 30 years (Bradford and Kastendick 2010), with extensive clearcutting as the main method for regeneration (Pastor et al. 2005). This has shifted the landscape to even-aged stands of early successional species, mainly aspen and paper birch (*Populus tremuloides*, *Populus grandidentata*) with a loss of late successional, conifer-dominated or mixed conifer and hardwood stands (Mladenoff and Pastor 1993, Cornett et al. 2001, Pastor et al. 2005, Schulte et al. 2007).

While forests in the northern Great Lakes region have undergone drastic change in the past ~150 years, they still provide many important ecological and economic functions. The boreal, sub-boreal, northern temperate ecotones in Minnesota and other areas of the northern Great Lakes region support forest ecosystems that provide enormous economic benefits to local and regional communities, while also supplying critical ecosystem services. For instance, these forests function as a large carbon sink (D’Amato et al. 2011), support numerous and diverse biological communities (Bradshaw et al. 2009), provide wood products and fiber for the timber industry (D’Amato et al. 2009), secure wildlife habitat for many endangered and charismatic species (Heinselman 1996, Niemi et al. 1998, Moen et al. 2008), and afford countless recreational opportunities (Duveneck et al. 2014).

The geographic location of Minnesota's forests combined with a range of natural and anthropogenic disturbance regimes has resulted in a mosaic of forest types and associations (TNC 2011). Many of the native tree species in this region are at their northern or southern range limit which suggests that such ecosystems will be especially sensitive to changes in climate (Ravenscroft et al. 2010, TNC 2011, Duveneck et al. 2014). In addition to climate-related stressors, the relative evenness of forest structure across a landscape itself can increased susceptibility and severity of attacks by insects. For example, the long legacy of fire suppression and clear-cut forest management in this region has substantially altered the susceptibility of these forests to attack by the spruce budworm (*Choristoneura fumiferana*), a dynamic that is less severe in Canada where forest species are the same, but management and fire legacies are substantially different (Robert et al. 2015). In silvicultural prescriptions involving cross-ownership management strategies to emulate natural disturbance regimes in time and space have been suggested to promote more functionally resilient forest types. This has produced forested landscapes that are less susceptible large-scale insect infestations, which, in theory, may also dampen the spread of exotic insects and disease (Frelich and Reich 2010).

Forest management and conservation agencies are faced with important decisions on how to best manage these forest lands by balancing human commodity needs with ecosystem goods and services. Many of these important ecosystem services (e.g., clean water, prevention of soil erosion, pest control, pollination, carbon storage, and local ecosystem resilience and stability) are the result of natural processes from healthy forest systems with rich structural and compositional diversity (Thompson et al. 2011). Hence, there is substantial interest in silvicultural systems that more effectively emulate natural disturbance and increase biological diversity, structural complexity, and spatial heterogeneity in managed forests (Franklin et al. 2002, Drever et al. 2006).

Ecological forestry and retention harvesting

Twenty-five years ago a new forest management model – ecological forestry – was introduced in the northwestern United States in response to the over simplification of forest structure and composition from intensive wood production and the need to better manage ecological values such as biodiversity and wildlife habitat (Franklin 1989). Ecological forestry practices are treatments that emulate patterns of species composition and structure developed under natural disturbance regimes (Hanson et al. 2012), and differs from traditional silviculture in that it shifts the focus toward structures that are left behind (e.g., live trees, snags, downed logs) versus what is being harvested (Franklin et al. 1997). Specifically, the three main principals of ecological forestry are: (1) incorporating biological legacies (live trees, snags, downed logs) into harvest prescriptions, (2) managing stands to sustain or restore structural and compositional heterogeneity, and (3) allowing species-appropriate cut rotation periods (Franklin et al 2007).

Harvesting that aims to create residual stand structures that more closely resemble the structural outcomes of natural disturbance regimes is known as variable retention harvesting (Franklin et al. 1997). This flexible management practice is gaining wide popularity as a tool for achieving complexity in forest stands managed principally for timber (Gustafsson et al. 2012;

Lindenmayer et al. 2012). By creating or retaining forest biological legacies and heterogeneous stand structures, variable retention harvesting may help promote greater biological diversity, augment critical ecosystem functions, and improve resilience to disturbance (Franklin et al. 2007). Specifically, retained trees serve as “life-boating” refugia for organisms and functions from the old stand to the new stand and for increasing connectivity across the landscape (Hansen et al. 1995, Franklin et al. 1997, Halpern et al. 2005). Depending on specific ecological goals, residual trees in a variable retention harvest can be left in either dispersed or aggregated spatial patterns (Franklin et al. 2007).

Dispersed retention leaves structures that are evenly distributed over a harvest unit, similar to shelterwood harvests, while aggregate retention focuses on leaving small forest patches of variable size and shape within harvest units (Franklin et al. 1997). While both dispersed and aggregated retention broadly maintain structural complexity of forest stands, each has its own ecological advantages (Franklin et al. 1997). Dispersed retention is more appropriate where the goal is to provide microclimates for regenerating plants (Macdonald and Fenniak 2007), uniformly distribute coarse woody debris to the forest floor or mitigate soil erosion over the entire harvest unit (Franklin et al. 1997).

Aggregate retention, on the other hand, can provide habitat similar to an undisturbed forest (Halpern et al., 1999), and allows for an easier opportunity to maintain a broader variety of stand structural elements (snags, diversity of tree species, sizes, and conditions), canopy lavers, understory plant species and communities, and intact forest floor layers compared to dispersed retention (Franklin et al., 1997). While live residual trees are important, associated dead standing and downed debris also play a crucial role in the forest ecosystem, especially for woodpeckers and other cavity nesting species (Franklin et al. 2007)

Non-living residual structures play an important role by modifying microclimate conditions creating suitable environments for organisms to survive (Franklin et al. 2007). Seedlings of some tree species only regenerate on decaying wood (Bolton & D'Amato 2011, Marx & Walters 2008, Caspersen & Saprunoff 2005, Mori et al. 2004, Cornett et al. 2001). In northeast Minnesota downed woody debris (DWD) is critical for the regeneration of long-lived species, such as yellow birch (*Betula alleghaniensis*) and northern white cedar (*Thuja occidentalis*), that were once dominant forest components but have declined since European settlement (Cornett et al. 2001, Bolton and D'Amato 2011). In one study, tree regeneration surveys showed that yellow birch seedlings and saplings occurred exclusively on DWD compared to that of the forest floor (Bolton & D'Amato 2011). This is directly related to the unique moisture and germination temperature requirements of this species, each of which are generally higher at critical times during the growing season on DWD logs and stumps (in more advanced stages of decay) than on the forest floor (Fraver et al. 2002). Similarly, decaying conifer logs serve as an important seedbed for northern white cedar (Cornett et al. 2001). In addition to providing suitable microenvironments for tree seed germination, DWD provides important substrates for fungi to colonize which provides a pathway for fungi dispersal throughout the forest (Hagan and Grove 1999). This is critical to forest health as many ectomycorrhizal fungi species associated with DWD have mutualistic relationships with living trees roots which assist the uptake of soil nutrients.

While downed logs provide long-term sources of moisture, nutrients and energy to the forest floor and soil (Krzyszowska-Waitkus et al. 2006), they also provide critical habitat for an array of vertebrate and invertebrate organisms (Franklin et al. 2000, Bunnel and Houde 2010), as well as suitable substrates for other regenerating vascular plants (Cornett et al. 2001, Bolton and D'Amato 2011). Moreover, DWD has an important influence on both hydrologic and geomorphic processes by trapping sediment, decomposing slowly, and mitigating soil erosion (Harmon et al 1986), providing functional elements of terrestrial and freshwaters ecosystems for many centuries (Franklin et al. 2000). For example, Fraver et al. (2002) found that even during exceptionally dry periods, DWD has a substantial water storage capacity, making it an extremely important structural feature for the forest by providing a large and relatively stable source of moisture when compared to leaf litter or mineral soil.

Together, standing dead wood (snags) and DWD provide valuable habitat for a wide array of wildlife species. In the northeastern United States scores of wildlife species use DWD and snags. For example, 18 species of mammal, 28 bird, 23 reptile and amphibian, and hundreds of invertebrate and fungi species utilize dead wood, either standing or fallen (Degraaf and Rudis 1986, Keddy and Drummond 1996). Downed woody debris is known to provide a moist microclimate for amphibians, refuge for small mammals, and, if large enough, for large mammals such as black bears *(Ursus americanus)* (Hagan and Grove 1999) and gray wolves *(Canis lupus)* (Bunnel et al. 2002). Downed logs are important habitat for the American (or pine) marten *(Martes americana*) and other members of *Mustelidae*, which tunnel under suspended logs during winter for traveling and resting (Hagan and Grove 1999). Sturtevant et al. (1997) states that marten do not inhabit areas lacking forest floor structure because they are not able to forage for subnivean (i.e., underneath snowpack) small mammals. Hence, if subnivean obligates are to inhabit and survive in managed forest areas, structurally diverse DWD must be retained and/or created, including logs and other DWD that is partially suspended above ground. Downed woody debris is also important for ruffed grouse, which utilize downed logs for their drumming courtship displays to attract mates (Hagan and Grove 1999).

Since its introduction in the 1980's, variable retention forestry has replaced clear-cut logging in many regions of the world (Vanha-Majamaa and Jalonen 2001, Beese et al. 2003, Gustafsson et al. 2010, 2012), is now a widely accepted tool for sustaining or restoring stand complexity in forests managed for timber (Gustafsson et al. 2012, LIndenmayer et al. 2012), and has increasingly been applied to global multifunctional forest ecosystems (Gustafsson et al. 2012). Due to the recognized value of late-successional forests for sustaining biodiversity and maintaining critical ecosystem services, including carbon storage, the focus of natural disturbance-based management has been to restore structural and compositional characteristics of late-successional forests to younger second-growth stands (Burton et al. 2009, Root et al. 2007, Keeton 2006). In Minnesota there is a growing trend toward decreased use of clearcutting as a management strategy and a concurrent increase in variable retention patch cutting, increased rotation ages, and retention of residual trees (D'Amato et al. 2009). Hence, it is clear that ecological forestry shows promise for restoring and maintaining structural and compositional diversity and increasing resilience and adaptive capacity (D'Amato et al. 2011).

Remote sensing potential

Despite over 25 years of scientific experimentation and practical application of retention forestry worldwide (Lindenmayer et al. 2012), there is little literature on using remotely sensed data for monitoring the structure and spatial patterns of retention trees and coarse woody debris within harvests treatments (Bater et al. 2009). Remote sensing is a tool that may potentially allow us to examine the spatial patterns and the distribution of structures (remaining live trees, snags, and DWD). Understanding the role and natural dynamics of DWD in forests is vital to improving forest management activities, and often requires long-term and repeated measurements on the same sites (Fraver et al. 2002). Organizations often lack the funding and time required to implement long term monitoring through multiple years of field data collection. Scientists, forest managers and conservation agencies would benefit from post-harvest monitoring of ecological change and to determine if both management and ecological goals were achieved. Little is known about how the retention of DWD or legacy trees to meet structural goals will impact regeneration (D'Amato et al. 2015). Having the ability to rapidly quantify DWD levels over large areas after harvest treatments are applied would provide an efficient way to determine if target retention levels were met and provide insight to specific areas where species of concern have good potential for regeneration. Thus, if satellite-based remote sensing technologies can be utilized for monitoring purposes it would benefit forest and resource management organizations by both improving the efficiency and frequency of monitoring activities and potentially providing insight to DWD distribution and volumes across the landscape.

Manitou collaborative forest management

The Nature Conservancy (TNC) has been collaborating with private and public landowners within the Manitou landscape (c.a. 42,000 ha) in northeast Minnesota (Figure 1) to promote and implement the principles of ecological forestry. The area consists of mainly fire-dependent forest communities and encompasses the entire watershed of Manitou River, Caribou River, and the East Branch of Baptism River (TNC 2009). Ownership is a mixture of private, county, state DNR, Superior National Forest, and TNC land. TNC has worked with the Minnesota DNR and Superior National Forest Service to develop and apply variable retention prescriptions on 490 ha (1,212 ac) and has conducted extensive ground-based ecological monitoring across ownership boundaries. TNC has many similar projects in this area and is interested in using remote sensing to more efficiently monitor and evaluate how well the retention harvests meet their management goals.

Goals and objectives

The primary focus of this research is to combine ground-based measurements of coarse woody debris and residual basal area with satellite remote sensing data to calibrate empirical prediction models for the purpose of producing spatially explicit estimates of these forest biophysical parameters. If successful, these satellite-based monitoring methodologies may serve as both a potential replacement for TNC's standard ground-based sampling protocols saving labor and time, and also provide a mechanism to facilitate more frequent monitoring assessments to guide adaptive forest management in northern Minnesota. Specifically, we addressed two primary research questions:

(1) Is it possible to calibrate estimation/prediction models for forest coarse woody debris by coupling ground-based measurement data with satellite image data (LANDSAT)?

(2) What respective accuracies can be expected among estimates of residual forest basal area and DWD using satellite image data (LANDAST) within harvest treatment areas?

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CHAPTER II. QUANTIFYING DOWNED WOODY DEBRIS AND RESIDUAL BASAL AREA FOLLOWING RETENTION HARVESTING IN NORTHEAST MINNESOTA USING LANDSAT SENSOR DATA

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Abstract

Restoration based forest management has increased significantly over the last decade across North America. Retention harvesting shows promise for restoring and maintaining forest structural and compositional diversity and also increasing resilience and adaptive capacity. This includes deliberate retention of large living trees, snags and downed woody debris (DWD). However, lack of consistent monitoring limits our understanding of the effectiveness of these strategies and our ability to adapt management accordingly. We investigate the use of readily available Landsat sensor data to remotely estimate and map DWD and basal area (BA) following retention harvesting in northeastern Minnesota, USA. We used multi-temporal winter Landsat throughout a single season to calibrate models for DWD (R^2: 0.54, RMSE = 19.02 m^3ha^{-1}), total BA (R^2: 0.55, RMSE = 1.85 m^2ha^{-1}), hardwood BA (R^2: 0.67 RMSE =1.23 m^2ha^{-1}), and conifer BA (R^2: 0.52 m^2ha^{-1}, RMSE = 0.94 m^2ha^{-1}). This novel approach uses winter imagery with varying snow accumulation to estimate and map residual forest structures. In addition to practical treatment monitoring, this research provides a valuable tracking tool from which we may deepen our long-term understanding of wildlife responses to DWD, fire and carbon dynamics, and forest nutrient cycling.

Introduction

Forest ecosystems in the northern Great Lakes region were significantly altered by land use change in the last 150 years since European settlement. These changes have resulted in unnatural simplification and homogenization of forest structure and composition at multiple spatial scales (Mladenoff and Pastor 1993, Schulte et al. 2007) and spatial patterns (White and Host 2008). Extensive logging (Zon 1925, Wolter et al. 2012) and intense slash fires (Stearns 1997) have diminished areas of unharvested forest and stands in the old-growth stage of development in Minnesota to c.a. 0.2% to 2% of their pre-European settlement extent, respectively (Frelich 1995). This has shifted the landscape from later successional forests dominated by conifers such as white spruce (*Picea glauca*), white pine (*Pinus strobus*) and northern white cedar (*Thuja occidentalis*) to an early successional landscape dominated by sprouting, shade intolerant hardwoods, mainly quaking aspen (*Populus tremuloides*) and paper birch (*Betula papyrifera*) (Schulte et al. 2007).

Many important ecosystem services (e.g., clean water, prevention of soil erosion, pest control, pollination, carbon storage, and local ecosystem resilience and stability) are the result of natural processes from healthy forest systems with rich structural and compositional diversity (Thompson et al. 2011). Unfortunately, traditional forest management of northern forests has focused on unnatural maintenance of particular forest states for economic benefit, while largely ignoring the dynamic complexities of ecosystem processes (Baskerville 1985, 1988). Increasingly over the last decade in North America, forest management and conservation agencies are faced with important silvicultural decisions on how to apply treatments that meet the economic demand for wood fiber while also promoting restoration of essential ecosystem services degraded by past forest management (Franklin 2003). One way to achieve this balance is by applying ecological forestry to younger, second growth stands to eventually restore and maintain structural and compositional characteristics of late-successional forests (Burton et al. 2009, Root et al. 2007, Keeton 2006).

Ecological forestry aims to create residual stand structure that more closely resembles natural disturbance and has become a widely used tool for achieving complexity (Gustafsson et al. 2012, Lindenmayer et al. 2012) and shows promise for restoring and maintaining structural and compositional diversity and increasing resilience and adaptive capacity (D'Amato et al. 2011). This includes the deliberate retention of larger diameter living trees, snags and, especially, downed woody debris (DWD) using group selection and irregular shelterwood harvests to restore historically diverse structural and compositional conditions (Hanson et al. 2012, Klingsporn et al. 2012, Keeton 2006). Residual live trees, snags, and downed woody debris or "biological legacies," are key post-harvest elements that help create heterogeneous stand structures that help promote biological diversity, critical ecosystem functions, and resilience to disturbance (Franklin et al. 2007, Robert et al. in review).

The ecological benefits of DWD are extensive. Downed woody debris provide not only long-term sources of nutrients and energy to the forest floor and soil (Krzyszowska-Waitkus et al. 2006), but have an important influence on hydrologic and geomorphic processes by trapping sediment, decomposing slowly, and controlling soil erosion (Harmon et al 1986). Fraver et al. (2002) found that even during exceptionally dry periods, DWD had substantially greater water storage capacity compared to adjacent leaf litter or mineral soil, making it an extremely important structural feature by providing an abundant, relatively stable source of moisture for plants, invertebrates, fungi, and moisture-sensitive amphibians. Moreover, presence of DWD is a critical seedbed for the regeneration of long-lived tree species that were once dominant forest components, but have since declined subsequent to European settlement, such as yellow birch and northern white cedar (Cornett et al. 2001, Bolton and D'Amato 2011). For instance, yellow birch research has shown that seedling establishment often occurs exclusively on DWD compared to that of the forest floor (Bolton and D'Amato 2011). Consequently, lower historical volumes of residual decaying logs for seed beds in northeast Minnesota is a strong factor limiting the spatial distribution and abundance of these two tree species (Cornett et al. 2001, Bolton and D'Amato 2011).

Research on the temporal trends of DWD across North America has shown that both volume and biomass of DWD in a forest generally follows "U-shape" pattern along a successional gradient; in that there is a moderate to high volume of DWD in early stands soon after disturbance, low levels in maturing stands, and the highest DWD volume occurs in late succession or old-growth stands (Sturtevant et al. 1997, Clark et al. 1998, Herbeck and Larsen 1999, Spetich et al. 1999, Idol et al. 2001). In fact, Hale et al. (1999) found that when comparing mature and old-growth characteristics of managed temperate hardwood forests in Minnesota, total DWD and standing dead tree volume was the only variable that was significantly indicative of old-growth forest condition of the seven independent variables tested (live basal area, sapling density, large seedling density, small seedling density, downed log volume, snag volume, and total DWD and standing dead tree volume). Generally, woody debris levels are high in old-growth stands as a result of past disturbance legacies. Over time, this residual debris decays slowly with little input from the regenerating stand. However, as the stand matures, it begins to accumulate DWD via natural tree mortality within the maturing stand from competition and small-scale disturbance, such as windthrow (Fraver et al. 2002, Sturtevant et al. 1997).

The previous stand and the current stand are the two main sources of extant DWD. The relative contributions from these sources differ with stand age over time (Spies et al. 1988), wherein DWD volume within early successional stages of forest development is almost entirely dependent on the previous stand (Spies et al. 1998): pre-disturbance debris, disturbance-generated debris, and residual standing trees that eventually die and contribute to the DWD pool (Sturtevant et al. 1997). Working in 60-80 year-old Douglas fir (*Pseudotsuga menziesii*) stands in western Oregon and Washington, Spies et al. (1988) found that pre-disturbance DWD accounted for c.a. 76% of total DWD observed.

Harvesting activities have significant impacts on residual snag and DWD volume and distribution long into the future (Morrissey et al. 2014, McGee et al. 1999, Duvall and Grigal 1999). For example, for nearly 50 years after partial harvesting the effects of this disturbance on the type, amount, distribution, and connectivity of DWD remained evident in *Quercus* dominated deciduous forests in Indiana (Morrissey et al. 2014). Moreover, even a single thinning was found to affect the distribution of DWD for several decades (Duvall and Grigal 1999). Harvesting typically removes what would otherwise be large diameter DWD (Bader et al. 1995, Morrissey et al. 2014), which has the effect of increasing the volume of smaller diameter logs or “slash” (Fraver et al. 2002).

In the Lake States, management of aspen for pulpwood has led to rotation ages of ± 30 years (Bradford and Kastendick 2010). Eventually, over many harvest rotations, DWD accumulation would be significantly reduced or cease all together (Duval and Grigal 1999), as only small amounts of DWD would be generated by the current stand. Stocks of DWD would then be limited to residual volumes left over after harvest, such as slash and declining trees (Sturtevant et al. 1997), likely to undermine silvicultural treatments aiming to create old-growth structure and encourage late successional species, and wildlife species dependent on DWD for habitat. Since forests of the Lake States largely lack older growth forest components (Frelich 1995), the effects of such management on DWD type and volume may be exacerbated, especially in young (0-30 yrs) forests (Duvall and Grigal 1999).

Managing for complexity and multiple values in a time of uncertain global environmental change may require a flexible approach to maintaining functioning ecosystems (Mladenoff and Pastor 1993, White et al. in review). Structural complexity, species diversity, and landscape scale spatial patterns are all components of complexity that are strongly influenced by management and important for maintaining and restoring resilient forests (Cornett and White 2013). Adaptive management is a key component to forest management strategies, allowing for management to be shifted in response to changing conditions and management outcomes (White et al. in review); this requires knowledge and understanding of forest change and responses to management and stressors (Deluca et al. 2010). In order to effectively manage for complexity, monitoring that captures key elements of structural complexity influenced by management at multiple spatial scales is necessary (Cornett and White 2013). However, the current lack of efficient and consistent monitoring efforts limits our understanding of the effectiveness of applied strategies and our ability to adapt efforts in a timely manner to more closely meet management goals (Deluca et al. 2010, Lindenmeyer et al. 2011). It may be possible to utilize remote sensing data for these monitoring purposes.

Organizations often lack the funding and time required to implement long-term monitoring through multiple years of field data collection. Being able to remotely quantify DWD levels after harvest treatments would not only provide an efficient way to determine if target levels were met for adaptive management, but potentially provide insight to potential wildlife habitat and specific areas where species of concern have good potential for regeneration. Restoration objectives often focus on achieving structural and composition elements of late-successional forests in second-growth forests (Root et al. 2007, Keeton 2006) and increasing historically important species (Crow et al. 2002). However, little is known regarding how the retention of coarse woody debris or legacy trees to meet late-successional structural objectives will impact regeneration development (D'Amato et al. 2015). Being able to quantify the distribution and abundance of residual legacy trees and DWD soon after retention treatments is invaluable for managers and agencies in order to understand if structural objectives were achieved, adjust accordingly, and drive future management decisions. There is a great need for empirical estimates of DWD following harvesting treatments (Klockow et al. 2013) and examination of the impacts of late-successional restoration treatments on the structural and compositional development of second-growth northern hardwoods (D'Amato et al. 2015).

The goals of our study were to examine if Landsat imagery could be used to quantify (1) DWD volumes and (2) residual live basal area (BA) following variable retention harvest treatments in mixed hardwood and conifer forest ecosystems. With respect to our first goal, we hypothesized that multi-temporal winter Landsat satellite imagery with high and low snow depths can be used to quantify volumes of large diameter DWD (≥10 cm diameter). Secondly, we hypothesize that winter Landsat sensor data with sufficiently high snow depth can also be used to quantity the characteristically sparse residual forest BA associated with these silvicultural treatments,. Previous multi-temporal studies, under substantially higher BA conditions, have shown that winter satellite imagery with snow ground cover produced stronger predictors of forest basal area than similar imagery from other seasons (Franco-Lopez et al. 2001, Wolter et al. 2008, Wolter et al. 2012). Moreover, winter Landsat imagery have also been used to estimate and map stands of medium BA, mature oak trees in within woodlands and savannas in central Minnesota (Wolter et al. 2012). The key advantage with snow ground cover is that it effectively covers small branches and leaf litter, soil, rock and other material on the forest floor that would otherwise confound the spectral signatures of standing forest trees (Brown et al. 2000, Chen & Cihlar 1996, White et al. 1995, Wolter et al. 2012). In doing so, snow ground cover provides a bright, spectrally homogenous surface upon which dark contrasting shadows of standing tree structures are accentuated (Seely 1949, Wolter et al. 2012). Moreover, when snow ground cover is thin (< 10 cm), individual horizontal elements of DWD directly on or held above the forest floor cast their own characteristic shadow patterns on snow (pers. obs.). However, as snow depth increases, shadow fraction of these DWD components changes disproportionately compared to vertical, residual forest elements. And, if snow accumulation continues, horizontal DWD shadow fraction is eliminated altogether, leaving only the shadow fraction of the vertical forest components. Given this, it follows that it may be possible to use two or more winter season Landsat images –with varying snow depths— to model both spatially explicit estimates of DWD volume per unit area as well as standing residual forest BA.

To our knowledge, this is the first study to attempt quantification of both DWD and residual forest basal area using multi-temporal winter satellite sensor data. In this study, we identify Landsat-based predictors to facilitate modeling DWD volume and residual forest BA, then we examine the degree to which these remote sensing predictors explain observed values in both DWD and residual forest BA. We employ xPLS regression (Wolter et al. 2012, Sing et al. 2013) with field plot data and multi-temporal Landsat sensor data to build models of DWD and residual forest BA.

Methods

Study site

The study site is approximately 490 ha (1,212 ac) within the Manitou Landscape located in Lake County in northeast Minnesota, USA (Figure 1). This area lies within the North Shore Highlands subsection of the Minnesota Ecological Classification System, which parallels the shore of lake Superior about 32 to 40 km (20 to 25 miles) inland. Elevation ranges from 200 to 700 m across this gently rolling landscape punctuated with some steep areas and bedrock outcroppings (MN DNR 2003). As part of the Cabin Lake Till Plain, soils are typically moderately well-drained, sandy loams to silt loams (SNF 2004). Lake Superior moderates the local climate throughout the year, resulting in cool, moist conditions in the spring and summer and warmer conditions in the fall and winter relative to inland areas (Baker and Keuhnast 1978, Baker et al. 1985). The overall continental climate has a mean growing season length of 104-168 frost-free days (base temperature = 0^{o}C), mean annual temperature of 4.7^{o}C, and mean annual precipitation of 77.5 cm with 150.4 cm of snowfall (1971–2000, Midwest Regional Climate Center, http://mcc.sws.uiuc.edu).

The study site lies within a complex mosaic of forest and wetland communities (MN DNR 2003). The upland forest is composed of northern mesic mixed forest, including various mixtures of quaking aspen, paper birch, balsam fir (*Abies balsamea*), white spruce, white pine, and northern white cedar (MN DNR 2003).

Pre-European settlement (c.a. 1850), upland forest vegetation was dominated by mixtures of eastern white pine, balsam fir, white spruce, northern white cedar, paper birch, and quaking aspen (White 2012). Northern hardwood patches occur on loamy uplands composed of sugar maple (*Acer saccharum*), yellow birch (*Betula allegheniensis*) and northern white cedar within the boreal conifer-hardwood matrix (White and Host 2000). Both white pine (9–19% to 0.2–1.0%) and white cedar (6–11% to 3.2–4.2%) have declined significantly from the pre-European settlement period to present in the North Shore Highlands subsection (White, 2001). Almost the entire area remains forested, with recreation and forest management the major land uses (MN DNR 2003). Stand replacing fire was an important disturbance (Heinselman 1973) and spruce budworm defoliation was and continues to be a significant disturbance to stands of balsam fir and spruce (MN DNR 2003, Robert et al. 2012).

Field data collection

We randomly established 34 field sampling plots throughout harvest treatments, with *in situ* measurement of residual forest components and regeneration occurring between 5 June and 14 August 2014. Field plots within treatment areas that were sampled in this effort were generated using the “Create Random Points” function in ArcMap 10.1, and located using a WAAS-enabled GPS receiver (Garmin GPSMAP 62stc; 2drms ≤ 3 m).

Each ground plot consisted of a total of five subplots: one located at plot center and four arranged orthogonally 30 m from plot center (Figure 3). The outer four subplots were spaced 30 m from plot center to facilitate integration with 30 m Landsat sensor data (Wolter et al. 2008). Subplot 1 was plot center (marked with a chaining pin), and a metric measuring tape was used to locate subplots 2 through 5 (each 30 m from plot center). A randomly assigned compass azimuth was used to navigate from subplot 1 (plot center) to the center of subplot 2. Subplots 2 through 5 were then separated by 90° around plot center (Figure 3).

At each subplot’s residual BA by species (live and dead) was collected using a metric basal area factor (BAF) one prism where plot radius varies according the bole diameter of each tree (Grosenbaugh, 1952). Diameter at breast height (DBH, 1.37 m above ground) was measured for each tree in the subplot. Bole diameter (cm) and species were recorded for all residual standing live and dead trees. In some cases standing dead trees had no bark and, hence, species was not readily discernable. Such dead trees were simply recorded as either dead hardwood or conifer. Two representative live tree heights per plot were measured near plot center using a clinometer, and the heights of all snags were also measured via ocular estimation-- including snapped-off trees. Residual basal area data (m^2ha^{-1}) collected at the five subplots were averaged to provide one set of values for each full plot. In addition to total basal area, separate hardwood and conifer basal area values were calculated for each plot to use as individual response variables. A summary of plot basal area by species is shown in Table 1.

At each plot, downed woody debris was sampled along two orthogonal 60 m transects and four 15 m transects between each subplot, totaling 180 m at each plot (Figure 4). Relatively short transect lengths (<100 m, Harmon and Sexton 1996) may limit the ability to fully capture variation in downed DWD at sites (D'Amato et al. 2008, Fraver pers. comm.). Howard and Ward (1972) found that the amount of downed wood was highly variable in harvested areas and that small number of long transects did not adequately pick up this variation; however, many shorter transects provided more reliable estimates of downed woody debris (Waddell 2002). For this reason, four 15 m transects were added between each subplot (180 m total transect length at each plot) to avoid over sampling near plot center and facilitate spatial integration with 30 m Landsat pixels. Only logs and branches encountered along transect greater than five cm in diameter were measured. The diameters were measured at the points of transect intersection for all DWD (Brown 1974). While 7.5 to 10 cm is a common minimum diameter in DWD assessments (Woodall et al. 2009), five cm was chosen due to the abundance of DWD with a diameter below 7.5 cm. Then, DWD was measured only if the transect crossed the center or central axis of the log (Brown 1974). If the transect crossed the same piece of DWD more than once, then it was measured at each intercept (Brown 1974). At each intercept DWD diameter (cm) and height above soil (cm to bottom of DWD) were measured.

Volume (m^3/ha) was estimated using the following formula (VanWagner 1968):

$$V = \left(\pi^2 \, \Sigma \frac{d^2}{8L}\right) X \frac{10{,}000 m^2}{ha}$$

Where *V* is wood volume, *d* is DWD diameter (m) and *L* is transect length in meters (van Wagner 1968). In line-intersect sampling theory, length (*L*) is considered to be one long sampling line at each plot (Hazard and Pickford 1979). While multiple DWD transects were used, the total transect length *(L*) remains as the total length of the line, which, in this case, is the sum of the lengths of all transects at each plot (Waddell 2002). Volume calculation, then, is independent of individual DWD piece length and size of the sampled area (VanWagner 1968).

While the Brown (1974) transect method of CWD sampling has become widely accepted as a standard inventory of CWD in the ecological community, there are three main sources of error that involve basic assumptions about log shape, log orientation relative to local topography and height of log relative to ground surface. First, relying on one diameter at the point of intersection to calculate volume assumes that log shape is cylindrical. This assumes logs have no taper and that the point of intersection represents the midpoint (VanWagner 1968). Fortunately, the CWD encountered in this study was logging residual, composed of pieces that were both cylindrical and straight. Second, pieces are assumed to be randomly oriented throughout a sample area with a Poisson distribution (Waddell 2002). If log residue is positioned primarily in one direction, then this assumption is violated and the sample is not considered random. This orientation bias can be substantially reduced by running two or more transects out from a common point at different angles (Van Wagner 1968, Baily 1970, Howard and Ward 1972, DeVries 1986, Hazard and Pickford 1986). By sampling along transect lines laid in different directions, as performed in this study, the number of logs intersected can be assumed unbiased and unrelated to their angle position (Waddell 2002). The final assumption is that all pieces lay horizontally on the ground; however, Van Wagner (1968) found that the vertical angle of a piece can be very large before a serious error will occur in the volume estimate. Logs encountered in this study were generally orientated horizontally with minor vertical angle (personal obs.)

Multiple DWD volumes were calculated and used as separate response variables. DWD volumes were calculated for logs with three specific lower limit diameter cut offs (≥ 5 cm, ≥ 7.5 cm and ≥ 10 cm) and for height ranges of DWD logs suspended above the forest floor (Table 6). This was done to explore model sensitivity; specificity, to test the hypothesis that Landsat data may only be able to sense larger debris (≥ 10 cm) on the forest floor. Additionally, these diameter limits are commonly used in DWD inventories, and knowledge on the ability and accuracy of modeling these specific diameter classes may prove useful for future work and research. Three additional DWD volumes were calculated using the diameter classes previously mentioned, but only for pieces ≥8 cm above the forest floor. DWD pieces that are lying on or partially embedded in the soil may be completely covered by relatively light snow. Creating a set of response DWD volume variables that specifically exclude that volume allows us to test weather DWD models improve by removing the volume that may potentially be cover by light snow. Modeled snow accumulation for the two low snow depth images was ~7.6 cm, hence, 8 cm was chosen as a height cut-off. In total, we calculated six DWD volumes, using both diameter and height classes and used these as sepearte response variables.

Landsat data

We used three Landsat-8 Operational Land Imager (OLI) images from the winter season immediately following the 2014 summer field data collection for this study, including November (N), December (D), and March (M) images (Table 2). All three images were downloaded from the USGS Earth Resources Observation and Science Center (EROS) Earth Explorer website (source: http://earthexplorer.usgs.gov/) in UTM zone 15 coordinates. These OLI images are made available in precision-orthorectified and radiometrically corrected format. Snow depth conditions among these three images range from 7.6 – 106.7 cm (Table 2). Modeled snow depth data was acquired from the Minnesota Climatology Working Group archive (source: http://climate.umn.edu/doc/snowmap.htm). To validate the accuracy of these modeled snow depth data, we collected actual snow depth information from two spatially disparate locations with varying snow depths in northeastern Minnesota on 19 January 2014 (Figure 5). Snow depths were recorded at 32 random points (point spacing >30 m to avoid fine-scale spatial autocorrelation) at each location then averaged and compared to the corresponding reported snow depth for 19 January 2014 from the modeled Minnesota snow map.

In addition to the selected satellite sensor's reflective bands for each image date (OLI 1–7), we derived several spectral indices to be used as candidate predictor variables for estimating CWD and residual BA. These indices included the commonly used normalized difference vegetation index (NDVI, Rouse et al. 1974), shortwave infrared-based (SWIR, OLI6 and OLI7) indices (moisture stress index, MSI, Rock et al, 1986; normalized difference snow index, NDSI, Dozier 1986; shortwave infrared visible ratio, SVR, Wolter et al. 2008), and the shortwave infrared band 6 to visible ratio, SVR6. Shortwave-IR indices were included in these analyses (tables 2 and 3) because formulations using SWIR wavelength intervals are known to be sensitive to forest BA (Horler & Ahern, 1986, Olsson 1994, Wolter et al. 2008), and have been used to study forest structural parameters, particularly forest density and tree size (Cohen & Spies, 1992, Cohen et al., 1995, Hansen et al., 2001, Lu et al., 2004). When compared to visible wavelengths, near-infrared (NIR) and SWIR regions of the electromagnetic spectrum experience relatively minor water vapor and Rayleigh scattering effects (Larsen & Stamnes 2005, Liang et al. 2002). Disproportionate diffuse irradiance (Dubayah 1994) in the visible region of the electromagnetic spectrum accounts for partial illumination of geometric shadows, which potentially lowers contrast sensitivity between the fully illuminated and shaded forest floor (Wolter et al. 2012). Hence, indices composed of NIR, SWIR, or contrasts between these and visible wavelengths enable the clearest possible contrast between sunlit and shaded forest floor signatures, and, hence, empirical relationships with forest BA (Wolter et al. 2012) and other structures casting shadows on the forest floor, such as downed woody debris. We included normalized difference shortwave infrared to green ratio (ND53) in these analyses for this reason. Visible bands (especially red) are known to be responsive to vegetation biomass (Roy & Ravan 1996) and other structural properties (Brown et al. 2000, Goetz & Prince 1996, Tucker 1979, Turner et al. 1999). Coniferous forest reflectance in the visible region varies inversely with biomass parameters such as basal area (Franklin 1986). The ND43 ratio (normalized difference red to green) was tested as a visible band index that may exhibit wavelength-specific variation in snow reflectance saturation that varies with forest structure (BA and DWD, see Dozier 1989).

Using winter Landsat data for forest structure mapping can be difficult due to the effect of low sun angle illumination on dissected terrain (Wolter 2008). However, the terrain within the study area is gentle and likely had negligible effects. To test for the potential effect of local terrain on geometric shadows (or hard shadows) or differences between sun-lit and hard-shaded surfaces, shaded relief images where created and included as explanatory variables. Shaded relief variables were created using a 30 m digital elevation model (DEM, source: http://ned.usgs.gov/) and solar ephemeris information that corresponds to the Landsat overpass time and date for each image.

We calculated and included difference values as explanatory variables for all seven reflective bands and seven indices by subtracting image values with high snow depth (March) from image values with low snow depth (November and December). High snow accumulation covers DWD that is present on the forest floor leaving solely the shadows of residual standing trees; while light snow cover allows for horizontal forest structure such as DWD on or above the forest floor to cast shadows on the snow. Calculating the shadow differences of multi-temporal images throughout a single winter season with high and low snow accumulation may provide unique a way to isolate and capture the shadows cast by DWD present on and above the forest floor. BA models were calibrated with a single high snow accumulation image (March) and both high and low snow accumulation images (March and November) to test the effectiveness of using single high snow accumulation image to cover forest floor structure.

Model calibration

Model calibration for each of the forest biophysical response variables (residual BA and DWD volume) using Landsat-based predictor variables (e.g., reflectance bands and derivatives) begins with predictor variable selection. However, because of the large potential number of possible image-based predictor variables, it is necessary to select the most parsimonious set of these predictors to avoid model over fitting (Babyak 2004). Here, this is performed automatically using iterative exclusion partial least squares (xPLS) regression (Wolter et al. 2012, Singh et al. 2013). This regression method is used to find a reduced set of explanatory variables that best fits single or multiple response variables without losing predictive precision (Wolter et al. 2012).

When predictor variables are relatively few, not significantly redundant, and have understood relationships to the response variable, multiple linear regression (MLR) is an appropriate modeling approach; if not, MLR is inappropriate or ineffective (Tobias 1995, Wolter 2012). Partial least squares (PLS) regression is an attractive option for building predictive models of response variables when there are many, highly collinear predictor variables (Geladi & Kowalski 1986, Wold et al. 2009), as is the case with remote sensing-derived predictor variables (Ingebritsen and Lyon 1985). Originally developed to calibrate models of response variables when predictor variables are numerous compared to sample size, PLS deals with collinearity by using latent variable structures (Geladi & Kowalski, 1986, Wold et al. 2009, Carrascal et al. 2009). However, while PLS regression reduces the weight of weak explanatory variables, it does not specifically exclude them, which results in unnecessarily large, cumbersome models (Wolter et al. 2008). Hence, xPLS differs from standard PLS in that xPLS systematically excludes predictor variables which exhibit little or no sensitivity to the response variable. The PLS regression approach extracts relatively few latent predictors (X-scores) and latent responses (Y-scores) from respective X and Y data matrices to indirectly predict the original set of response variables (Wolter 2012). This mitigates unstable collinear effects in both X and Y data space (Helland, 1988), while only assuming that relationships between X and Y are linear (Wold et al., 2009). Unlike principal components regression (PCR) where X-scores are extracted from an X-matrix of predictors (spectral decomposition of X'X) (Massy, 1965), PLS regression is more specific by involving singular decomposition of X'Y (predictor and response variables) (Wolter 2012). By doing so, directions in latent variable space (associated with high variance in the response) are selected to maximize the relationship strength between consecutive pairs of scores (Geladi &Kowalski 1986, Tobias 1995).

In the xPLS routine, every potential explanatory variable is excluded from the model development once and returned to the pool of potential predictors until all such variables have been evaluated. The one excluded predictor variable that resulted in the best model (lowest RMSE of prediction) is then permanently discarded from the pool of predictor variables (Wolter 2012). This process is repeated until the RMSE of prediction can no longer be reduced. The resulting "best model" consists of the lowest number of image predictor variables and lowest root mean error of prediction (Wolter et al. 2012). In general, the PLS regression routine has been found to be a more reliable statistical approach than multiple regression and principal components analysis for identifying relevant explanatory variables, as well as their magnitude of influence in ecological studies (Carrascal et al. 2009), and the PLS hybrid routine, xPLS, is repeatable and has consistently produced streamlined models with high levels of parameter estimation accuracy (Wolter et al. 2008, 2009, 2012, Wolter & Townsend, 2011).

Pre-analysis data screening

It is well known that partial least squares regression has sensitivity to outliers (Rousseeuw and Leory 2003). To account for this sensitivity, the dependent variables in this research (BA and DWD) were analyzed to identify potential outliers and deviations from normality both visually and via Shapiro-Wilk tests using R-Studio. Some researchers recommend the Shapiro-Wilk test as the best choice for testing normality of data (Thode 2002), and is especially recommended when the sample size of data is less than 50 (Elliott and Woodward 2007). The Shapiro-Wilk test assumes a null hypothesis that the data likely come from a normal distribution. Thus, a p-value less than or equal to 0.05 suggests the data are not normally distributed (null hypothesis false), and a p-value greater than 0.05 suggests the data are normally distributed (no reason to reject null hypothesis). Hence, if present, outliers were removed and non-normal distributions transformed accordingly to achieve normality.

Results

Dependent DWD and BA variables

Total residual basal area was low throughout all plots (mean 6.7 $m^2\ ha^{-1}$, range 2.8-14.6 $m^2\ ha^{-1}$), with hardwood and conifer residual basal area measuring mean 4.6 $m^2\ ha^{-1}$ (range 1.4-10.4 $m^2\ ha^{-1}$) and mean 2.1 $m^2\ ha^{-1}$ (range 0.2-5.2 $m^2\ ha^{-1}$) respectively. DWD volume for all logs with diameters ≥ 5 cm averaged 67.8 m^3ha^{-1} (range 24.0-144.6 m^3ha^{-1}); ≥ 7.5 cm diameters averaged 62.9 m^3ha^{-1} (range 22.6-132.5 m^3ha^{-1}), ≥ 10 cm diameter averaged 55.2 m^3ha^{-1} (range 12.0-127.0 m^3ha^{-1}; Table 6). DWD volume in the ≥ 5 cm diameter and 8 cm off the ground category averaged 26.9 m^3ha^{-1} (range 4.8-73.5 m^3ha^{-1}), DWD volume in the ≥ 7.5 cm diameter and 8 cm off ground category averaged 22.6 m^3ha^{-1} (range 4.2-72.0 m^3ha^{-1}), and DWD volume in the ≥ 10 cm diameter and 8 cm off ground category averaged 22.1 m^3ha^{-1} (range 3.8 – 72.0 m^3ha^{-1}). Full descriptive statistics of dependent variable data are provided in Table 5.

Model development

All dependent variables were natural log transformed to achieve normality. Model predictions were back transformed and RMSE was calculated on the original dependent variable scale. Using the following formula:

$$RMSE = \sqrt{\frac{\sum_{l=1}^{n}(y_l - \hat{y}_l)^2}{n}}$$

where n is the number of observations, y_l is the observed value and $\hat{y}_l$ is the estimated value. Final dependent variable models were obtained via implementation of the PLS regression operation in SAS using the xPLS-reduced set of image predictor variables for each response variable. Downed woody debris models used difference values of three images with high and low snow depths (March-November and March-December) as predictor variables. This resulted in adjusted coefficients of determination (Adj R^2) of 0.43 m^3ha^{-1} (RMSE 22.70 m^3ha^{-1}), 0.54 m^3ha^{-1} (RMSE 19.02 m^3ha^{-1}), and 0.52 m^3ha^{-1} (RMSE 18.05 m^3ha^{-1}) for total DWD volume with diameters ≥ 5 cm, ≥ 7.5 and, ≥ 10 cm respectively. Model calibration for DWD suspended above the forest floor 8 cm above snow cover at resulted in adjusted coefficient of determination (Adj

R^2) of 0.53 m^3ha^{-1} (RMSE 10.49 m^3ha^{-1}) for DWD ≥ 10 cm diameter. There was no model fit for DWD ≥ 5 cm and ≥ 7.5 cm in diameter.

Modeled calibration of residual forest BA developed using single March image resulted in an adjusted coefficient of determination (Adj R^2) of 0.55 m^2ha^{-1} (RMSE 1.85 m^2ha^{-1}), 0.67 m^2ha^{-1} (RMSE 1.23 m^2ha^{-1}), and 0.49 m^2ha^{-1} (RMSE 1.00 m^2ha^{-1}) for total, hardwood, and conifer BA respectively. Modeled calibration of residual forest BA developed using both March and November images resulted in an adjusted coefficient of determination (Adj R^2) of 0.47 m^2ha^{-1} (RMSE 2.01 m^2ha^{-1}), 0.31 m^2ha^{-1} (RMSE 1.88 m^2ha^{-1}), and 0.52 m^2ha^{-1} (RMSE 0.94 m^2ha^{-1}) for total, hardwood, and conifer BA, respectively. Model calibration using single March image produced superior results for total BA and hardwood BA, where the conifer BA model was slightly improved using March and November variables compared to only March variables.

Best fit model in each category (DWD, total BA, hardwood BA, and conifer BA) was selected by highest Adj. R^2 and lowest relative RMSE. Fit plot of predicted versus observed values for best models is shown in Figure 6. Iterative partial least squares regression results (Table 7) show hardwood BA as retaining the highest number of predictor variables (8), followed by DWD ≥ 7.5 cm diameter (4), conifer BA (4), and total BA (3). DIFFSHD_MD, DIFF6_MN, and DIFF5_MD were retained in all DWD models. Best fit models were used to create structural estimate maps throughout the study area (Figures 7-10).

Discussion

The results of this study suggest that ground and remote sensing data may be used in combination to calibrate biophysical models of forest structure following retention harvesting in mixed hardwood and conifer forests of the Great Lakes region, USA. It is evident that winter Landsat satellite imagery can be used to produce accurate estimates of downed woody debris volume and live residual forest basal area. There is sparse research on estimating DWD using remote sensing technologies, particularly in the Great Lakes region of the USA. While not directly comparable due to different forest ecosystem and remote sensing technology used, the accuracy of our DWD model calibrations were less than that reported for a study where airborne LiDAR data was used to model DWD in oak dominated forest in central Appalachia, Kentucky, USA (van Aardet et al. 2011). However, our results are similar to those of Huang et al. (2009) where the authors used Airborne Synthetic Aperture Radar (AirSAR) data to calibrate DWD models in post-fire conifer dominated stands in Yellowstone National Park.

While the results of DWD modeling are encouraging, there all potentially multiple causes confounding model accuracy. Using difference values from high and low snow accumulation images does allow for the ability to extract forest structure on/and near the forest floor. While DWD is a major component of forest floor structure, shrubs are also present, and not accounted for. Additionally, conifer species with canopies near the forest floor can shade DWD present below or near the tree (pers. obs.). Windthrow is a common disturbance in the forests of northern Minnesota (Mladenoff and Pastor 1993). During field collection on multiple occasions, strong winds resulted in standing live trees falling to the ground (pers. obs.). This likely introduced error to both DWD and BA modeling as March imagery was acquired prior to data collection.

Previous studies have shown that multi-temporal winter imagery with snow ground cover produce stronger predictors of forest basal area than imagery from other seasons

(Franco-Lopez et al. 2001, Wolter et al. 2012, Wolter et al. 2008). Specifically, Landsat winter data –with snow ground cover-- has been found to be 90% effective in measuring basal area of mature trees in oak woodlands and savannas in central Minnesota (Wolter et al. 2012) with substantially greater BA than the low BA ranges modeled in this study. Unfortunately, our cross-validated accuracy results for total basal area (Adj R^2 = 0.55 m^2ha^{-1}; RMSE 1.85 m^2ha^{-1}), hardwood basal area (Adj R^2 = 0.67 m^2ha^{-1}; RMSE 1.23 m^2ha^{-1}), and conifer basal area (Adj R^2 = 0.52 m^2ha^{-1}; RMSE 0.94 m^2ha^{-1}) are less than that reported by Wolter et al. (2012). However, previous studies were conducted in areas with significantly higher total forest basal area (Franco-Lopez et al. 2001), often in homogenous forest stands that covered the entire plot area (Wolter et al. 2012, Wolter et al 2008), and used both leaf-on and leaf-off imagery as predictor variables. Our study area contained very few residual trees per plot (average total basal area 6.69 m^2 ha^{-1}) composed of at least three different species including at least one coniferous species (see Table 1).

Additionally, 13.6% of residual trees were snags with broken tops. Trees with broken tops do not cast the same amount of shadow onto the forest floor compared to a tree with full canopy that has an intact bole not snapped off (pers. obs). Because tree bole diameter is allometrically related to both tree height and volume (Jenkins et al. 2003), we explored such allometry that was developed for the Great Lakes area (Perala and Alban 1994) and general national-scale (Jenkins et al. 2003) as a potential means of correcting for snag heights with broken tops. Since DBH and height to break of all snapped off trees was known, allomtery could be then be applied to find the estimated height of a given tree and the used to scale the expected volume. Unfortunately, the Perala and Alban (1994) allometry is soil-specific, but excludes soil types specific to our study area. Jenkins et al. (2003) more general allomtery did not prove accurate when applied to our study area field data.

The focus of natural disturbance based silviculture in northern hardwood forests has been to restore the structures and processes that are characteristic of old-growth forests to younger second-growth stands (Burton et al. 2009). Structural diversity has been shown to be important for long-term productivity, habitat, resilience, and adaptive capacity (Hardiman et al. 2011, Lei et al. 2009). Extended rotations and recovery periods are often cited as an ecologically based method for promoting forest heterogeneity and structure (Morrissey et al. 2014); however, unless there is an active approach to create and manage CWD, creating old-growth structure is unlikely to be successful (Duval and Grigal 1999, Morrissey et al. 2014).

Harvesting activities significantly impact DWD volume and distribution long into the future (Morissey et al. 2014, McGee et al. 1999, Duvall and Grigal 1999). DWD within early successional stages after disturbance is almost entirely depended on the previous stand (pre-disturbance debris, disturbance-generated debris, and residual standing trees that eventually die and contribute to the DWD pool) (Spies et al. 1998, Sturtevant et al. 1997). Post-harvest DWD and snag levels may be extremely important for creating late-growth structure, creating favorable microsites for regenerating late-successional species, and providing wildlife habitat for CWD dependent species. Residual live trees also play important roles by “lifeboating” organisms and functions from old stand to the new stand (Franklin et al. 2007), and eventually dying and contributing to the DWD pool. Being able to remotely quantify spatially explicit DWD volumes and RBA after harvest treatments is valuable to for managers as a tool to aid monitoring activities and could provide ecological insights to management results.

In northern Minnesota, regeneration is poor for characteristic long-lived species (yellow birch, white cedar and white spruce), and much of the mature forest that is transitioning to later-successional growth stages is lacking not only key species but the structural characteristics needed to develop late-successional conditions (White et al. in review). Low volumes of decayed logs to provide seedbeds may limit white cedar and yellow birch regeneration (Cornett et al. 2000, Bolton and D'Amato 2011). These genera (*Betula*, *Picea*, and *Thuja*) primarily establish on CWD or exposed mineral soil seedbeds relative to the undisturbed forest floor (Cornett et al., 2001; Caspersen and Saprunoff, 2005; Shields et al., 2007; Marx and Walters, 2008). DWD may play an important role in providing higher surface temperatures when compared to the undisturbed forest floor. Much of the timber harvesting in this region occurs during winter months when the soil is frozen and covered by snow, limiting levels of exposed mineral soil (Shields et al., 2007); DWD may be the only suitable microhabitat for these long-lived historically important species within silviculture prescriptions that do not deliberately scarify the soil (Bolton and D'Amato 2011). Natural disturbance based prescriptions will do little to restore native tree diversity if deliberate creation of suitable microsites historically generated by natural disturbance (e.g., highly decayed wood, exposed mineral soil) are not included (Bolton and D'Amato 2011).

Forest DWD models for the Manitou harvest areas are not location specific, but do require imagery with different snow depths. However, northeast Minnesota does experience significant snowfall annually, with mean annual snowfall of 150.4 cm (1971-2000, Midwest

Regional Climate Center, http://mcc.sws.uiuc.edu). It is necessary to test how robust this approach may be for other areas and forest types where snow depth data is available. Successfully characterizing these relationships will pave the way for more efficient monitoring of key forest structure elements.

Conclusions

The results of this study suggest it is possible to remotely estimate downed woody debris following harvest treatments using free and widely available Landsat data. Agencies often lack the funding and/or time to carry out extensive field measurements, especially when monitoring at the landscape scale across ownership boundaries. These models provide managers, scientists, and organizations with a quick way to monitor post-harvest structure and distribution (CWD and residual basal area), potentially supplementing or replacing traditional labor-intensive field measurements. Not only does this provide a way to evaluate if target goals were reached after harvest treatments (specific density of residual trees / volume of downed woody debris), but insights to potential wildlife habitat.

Effectively managing for complexity requires monitoring that captures key elements of complexity that are influenced by management (Mladenoff and Pastor 1993, Cornett and White 2013). Adaptive management is a key component of forest management strategies allowing for shifts in management in response to changing conditions and management outcomes (White et. al). This requires knowledge and understanding of forest change and responses to management and stressors (Deluca et al. 2010). The ability to remotely quantify DWD volumes after harvesting treatments can be a key element in monitoring programs to capture finer forest structure, and drive future management decisions.

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Table 1. Relative BA by species at each plot; *Betula papyrifera* (BEPA), *Populus tremuloides* (POTR), *Picea sp.* (PICE), *Thuja occidentalis* (THOC), *Abies balsamea* (ABBA), *Acer rubra* (ACRU), *Pinus strobus* (PIST), snag with no bark and unable to accurately identify species (UNKWN), hardwood (HWD), and conifer (CON).

PLOT	BA (m^2/ha) Plot Avg	Rel_BA BEPA	Rel_BA POTR	Rel_BA PICE	Rel_BA THOC	Rel_BA ABBA	Rel_BA ACRU	Rel_BA PIST	Rel_BA UNKWN	% HDW	% CON	Rel_BA SNAG
N01	5.6		0.57	0.43						0.57	0.43	0.07
N04	6.4	0.34	0.47	0.16		0.03				0.81	0.19	0.44
N05	14.6	0.51	0.07	0.11	0.21	0.04	0.07			0.64	0.36	0.21
N08	8.2	0.44	0.07	0.39	0.05	0.05				0.51	0.49	0.39
N11	6.4	0.22	0.19	0.44	0.13	0.03				0.41	0.59	0.19
N15	4.0		0.65	0.20					0.15	0.65	0.20	0.20
N37	8.4	0.69	0.07	0.10	0.14					0.76	0.24	0.40
N38	8.8	0.34	0.41	0.18	0.02	0.02		0.02		0.75	0.25	0.23
W10	4.4	0.32	0.36	0.23	0.05	0.05				0.68	0.32	0.41
W14	5.4	0.81	0.11			0.04	0.04			0.96	0.04	0.56
W17	7.6	0.71	0.13	0.03	0.08	0.03	0.03			0.87	0.13	0.32
W18	5.2	0.58		0.08	0.27	0.08				0.58	0.42	0.50
W19	5.2	0.73		0.15	0.08			0.04		0.73	0.27	0.35
W20	4.0	0.70		0.10	0.05	0.15				0.70	0.30	0.15
W21	5.6	0.07	0.18	0.36		0.39				0.25	0.75	0.21
W22	6.0	0.40	0.03	0.33		0.13		0.10		0.43	0.57	0.17
W23	6.2	0.55	0.06	0.35		0.03				0.61	0.39	0.19
W24	6.0	0.73	0.10	0.07		0.07	0.03			0.87	0.13	0.23
W25	9.2	0.54		0.35	0.07	0.02	0.02			0.57	0.43	0.33
W26	7.0	0.31	0.43	0.20	0.03		0.03			0.77	0.23	0.37
W27	9.4	0.43	0.11	0.11	0.06	0.13	0.17			0.70	0.30	0.30
W28	11.4	0.32	0.25	0.35	0.02	0.05	0.02			0.58	0.42	0.14
W29	4.6		0.43	0.48		0.09				0.43	0.57	0.00
W3	4.8	0.25	0.29	0.38		0.04	0.04			0.58	0.42	0.38
W30	10.6	0.28	0.36	0.19	0.02	0.09	0.04		0.02	0.68	0.30	0.28
W31	5.6	0.39	0.46	0.14						0.86	0.14	0.39
W36	6.0	0.20	0.53	0.07	0.10	0.07			0.03	0.73	0.23	0.33
W41	2.8	0.64	0.21	0.14						0.86	0.14	0.64
W42	7.2	0.64	0.33			0.03				0.97	0.03	0.19
W43	6.6	0.85	0.09	0.03	0.03					0.94	0.06	0.45
W44	4.2	0.05	0.71	0.14			0.10			0.86	0.14	0.33
W45	11.4	0.58	0.14	0.07		0.02	0.19			0.91	0.09	0.37
W7	4.6	0.26	0.30	0.35		0.09				0.57	0.43	0.13
W9	4.0	0.35	0.10	0.40	0.05			0.10		0.45	0.55	0.40

Table 2. Landsat 8 images used, solar elevation, azimuth, and snow depth.

Image date	Image code	Solar elevation	Solar azimuth	Modeled snow depth (cm)
3/4/2014	M	33.41	155.09	107
11/15/2014	N	22.97	165.56	7.6
12/1/2014	D	19.55	165.09	7.6

Table 3. Landsat 8 bands used as explanatory varialbes and to derive indices and their respective region of the electromatnetic spectrum and wavelength (micrometers).

Landsat 8 Satellite Band	Electromagnetic spectrum region	Wavelength (micrometers)
1	Coastal aerosol	0.43 - 0.45
2	Blue	0.45 - 0.51
3	Green	0.53 - 0.59
4	Red	0.64 - 0.67
5	Near Infrared (NIR)	0.85 - 0.88
6	Shortwave Infrared (SWIR1)	1.57 - 1.65
7	Shortwave Infrared (SWIR2)	2.11 - 2.99

Table 4. Indices used to detect DWD volume and RBA include normalized difference snow index (NDSI), normalized difference of bands 5 and 3 (ND53), normalized difference vegetation index (NDVI), normalized difference of bands 4 and 3 (ND43), shortwave infrared visible ratio (SVR) , shortwave infrared band 6 visible ratio (SVR6), and moisture stress index (MSI).

Indices	Formulation using Landsat bands
NDSI	((3-6) / (3+6) +1) *100
ND53	((5-3) / (5+3) +1) *100
NDVI	((5-4) / (5+4) +1) *100
ND43	((4-3) / (4-3) +1) *100
SVR	((AVG:6,7) / (AVG:2,3,4) +1) *100
SVR6	((6) / (AVG:2,3,4) +1) *100
MSI	((6-5) / (6+5) +1) *100

Table 5. Descriptive statistics for dependent DWD and BA variables.

Response Variable	Average	Minimum	Maximum	Std. Dev.
Total Basal Area (m2 ha−1)	6.89	2.8	14.6	2.58
Hardwood Basal Area (m2 ha−1)	4.61	1.4	10.4	2.17
Conifer Basal Area (m2 ha−1)	2.05	0.2	5.2	1.36
DWD Volume (m3/ha) ≥ 5cm diameter	67.84	23.99	144.64	30.65
DWD Volume (m3/ha) ≥ 7.5 cm diameter	62.86	22.63	132.52	29.83
DWD Volume (m3/ha) ≥ 10 cm diameter	55.16	11.98	127.03	28.44
DWD Volume (m3/ha) ≥ 5 cm diameter and 8 cm height off ground	26.86	4.78	73.52	17.09
DWD Volume (m3/ha) ≥ 7.5 cm diameter and 8 cm height of ground	24.55	4.24	71.96	16.62
DWD Volume (m3/ha) ≥ 10 cm diameter and 8 cm height off ground	21.12	3.78	71.96	16.17

Table 6. Results of DWD and BA calibrations / validations.

Dependent Variable	n	Adj R^2	R^2	RMSE (original scale)	PRESS	p-value	B_o	B_1	Vars. Initial	Vars. Used	Source Image
lnDWD Volume (m3/ha) ≥ 5cm diameter	34	0.4309	0.4481	22.7045	0.7523	<0.001	2.27408	0.44811	30	6	M,D,N
lnDWD Volume (m3/ha) ≥ 7.5 cm diameter	**34**	**0.5361**	**0.5501**	**19.0202**	**0.7544**	**<0.001**	**1.81457**	**0.55013**	**30**	**4**	**M,D,N**
lnDWD Volume (m3/ha) ≥ 10 cm diameter	34	0.5193	0.5339	18.0451	0.7729	<0.001	1.80709	0.53390	30	6	M,D,N
lnDWD Volume (m3/ha) ≥ 5 cm diameter and 8cm height off ground	29	---	---	---	---	---	---	---	30	-	M,D,N
lnDWD Volume (m3/ha) ≥ 7.5 cm diameter and 8cm height off ground	29	---	---	---	---	---	---	---	30	-	M,D,N
lnDWD Volume (m3/ha) ≥ 10 cm diameter and 8cm height off ground	34	0.5301	0.5443	10.4916	0.7764	<0.001	1.27901	0.54433	30	8	M,D,N
lnTotal Basal Area (m2 ha−1)	**33**	**0.5495**	**0.5632**	**1.8464**	**0.718**	**<0.001**	**0.80171**	**0.56319**	**15**	**2**	**M**
lnTotal Basal Area (m2 ha−1)	33	0.469	0.4851	2.0110	0.7632	<0.001	0.94506	0.48508	30		M,N
lnHardwood Basal Area (m2 ha−1)	**34**	**0.6705**	**0.6804**	**1.2301**	**0.7169**	**<0.001**	**0.45285**	**0.68045**	**15**	**8**	**M**
lnHardwood Basal Area (m2 ha−1)	34	0.3122	0.333	1.8797	0.7259	<0.001	0.94525	0.33300	30	8	M,N
lnConifer Basal Area (m2 ha−1)	34	0.4879	0.5034	0.9951	0.8305	<0.001	0.50528	0.50342	15	3	M
lnConifer Basal Area (m2 ha−1)	**34**	**0.5211**	**0.5356**	**0.9433**	**0.7259**	**<0.001**	**0.09809**	**0.08816**	**30**	**3**	**M,N**

Table 7. Explanatory variables retained by xPLS regression for each DWD and BA model.

Dependent Variable	Explanatory image variables
lnDWD Volume (m3/ha) ≥ 5cm diameter	DIFF6_MN, DIFF5_MD, DIFFND43_MN, DIFFMSI_MM, DIFFSVR_MD, DIFFSHD_MD
lnDWD Volume (m3/ha) ≥ 7.5 cm diameter	**DIFF6_MN, DIFF5_MD, DIFFND53_MN, DIFFSHD_MD**
lnDWD Volume (m3/ha) ≥ 10 cm diameter	DIFF6_MN, DIFF5_MD, DIFFND43_MN, DIFFMSI_MN, DIFFND53_MN, DIFFSHD_MD
lnDWD Volume (m3/ha) ≥ 5 cm diameter and 8cm height off ground	---
lnDWD Volume (m3/ha) ≥ 7.5 cm diameter and 8cm height off ground	---
lnDWD Volume (m3/ha) ≥ 10 cm diameter and 8cm height off ground	DIFF6_MN, DIFF3_MD, DIFF4_MD, DIFF5_MD, DIFFND53_MD, DIFFND43_MD, DIFFSHD_MN, DIFFSHD_MD
lnTotal Basal Area (m2 ha−1)	**M2, M5, M6**
lnTotal Basal Area (m2 ha−1)	NDSI_M, MSI_N
lnHardwood Basal Area (m2 ha−1)	**M2, M5, NDSI_M, ND53_M, NDVI_M, ND43_M, SVR6_M, MSI_M**
lnHardwood Basal Area (m2 ha−1)	M1, M2, M4, M6, M7, N6, ND43_N, SVR_N
lnConifer Basal Area (m2 ha−1)	M1, ND53_M, NDVI_M
lnConifer Basal Area (m2 ha−1)	**ND53_M, NDVI_M, SVR6_N**

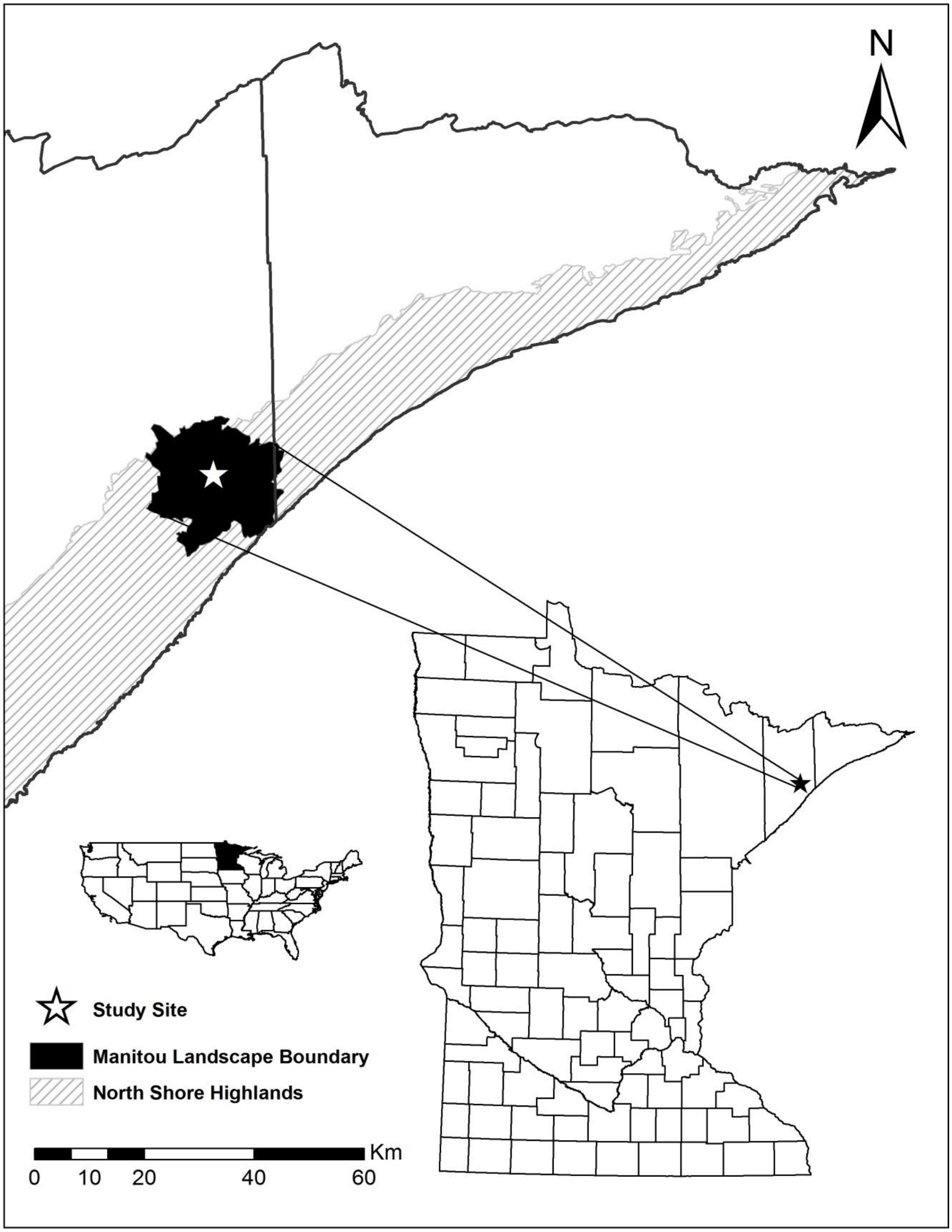

Figure 1. Study site location in northeast minnesota withing greater Manitou landscape and North Shore Highlands ecological subsection.

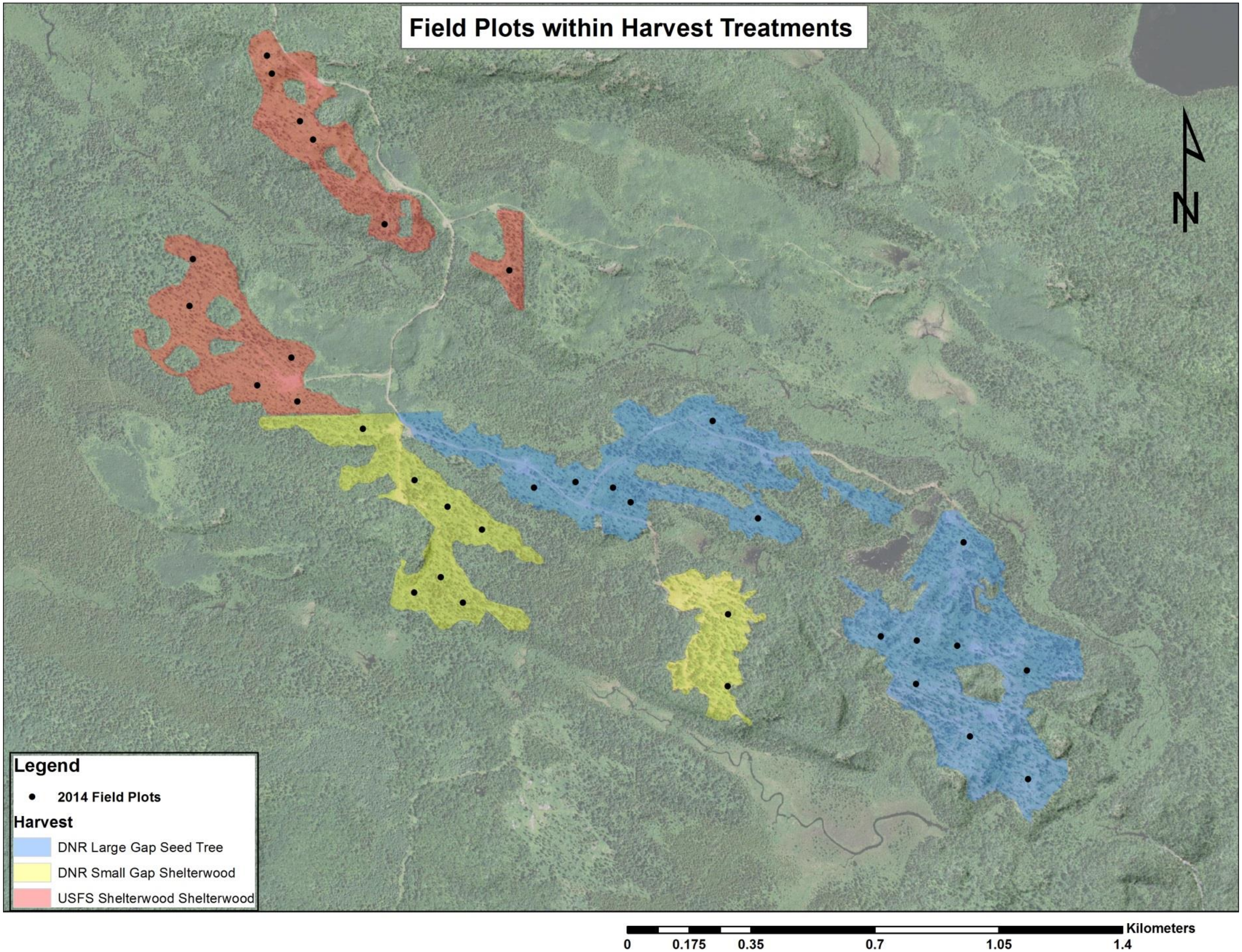

Figure 2. Field plots within retention harvest treatments.

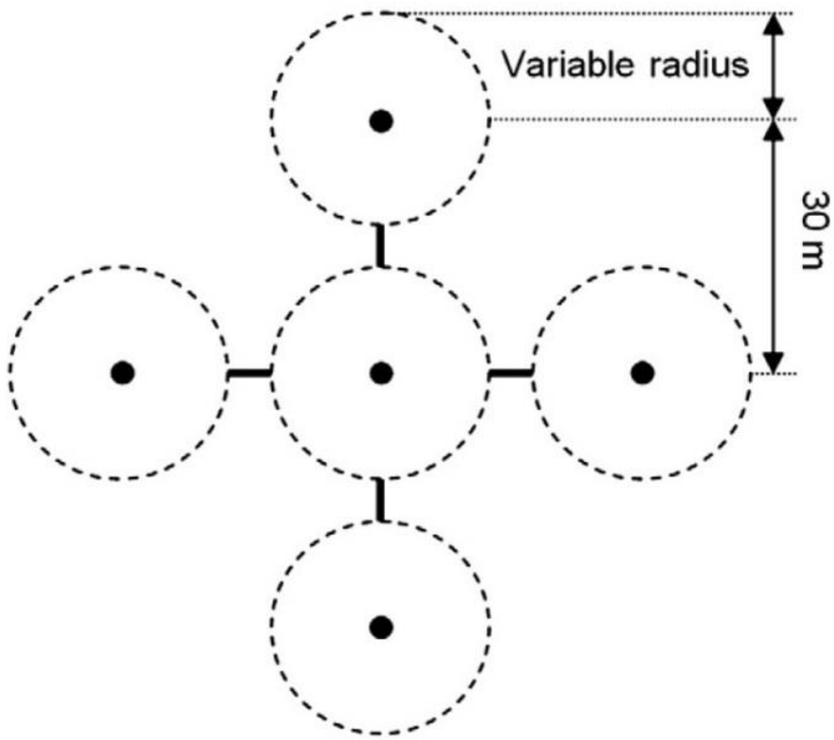

Figure 3. Single ground basal area plot with five variable radius subplots along two orthogonal axes (Wolter et al. 2009).

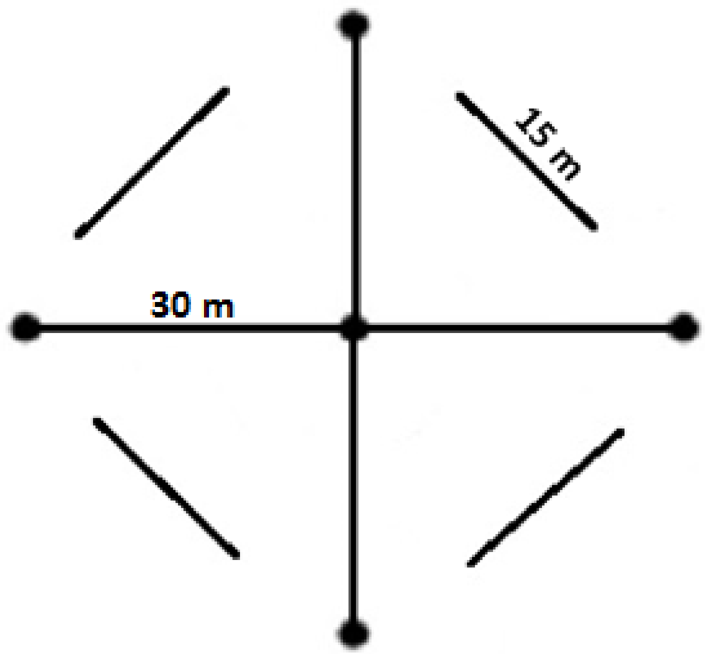

Figure 4. Single ground plot downed woody debris transect design.

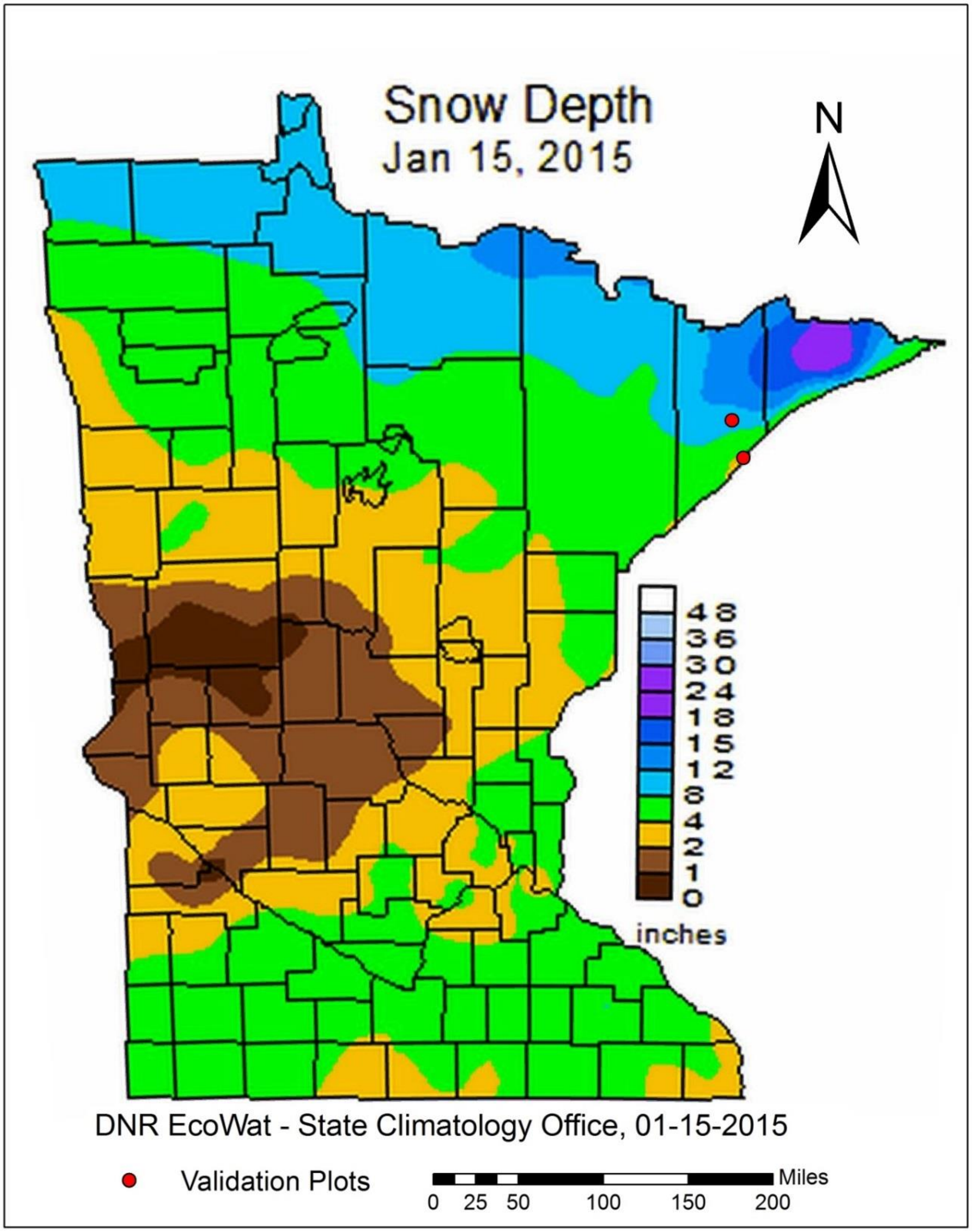

Figure 5. Map of statewide snow depth data acquired from Minnesota Climatology Working Group archive (source: http://climate.umn.edu/doc/snowmap.htm) and locations of validation plots to gauge accuracy of modeled snow depth.

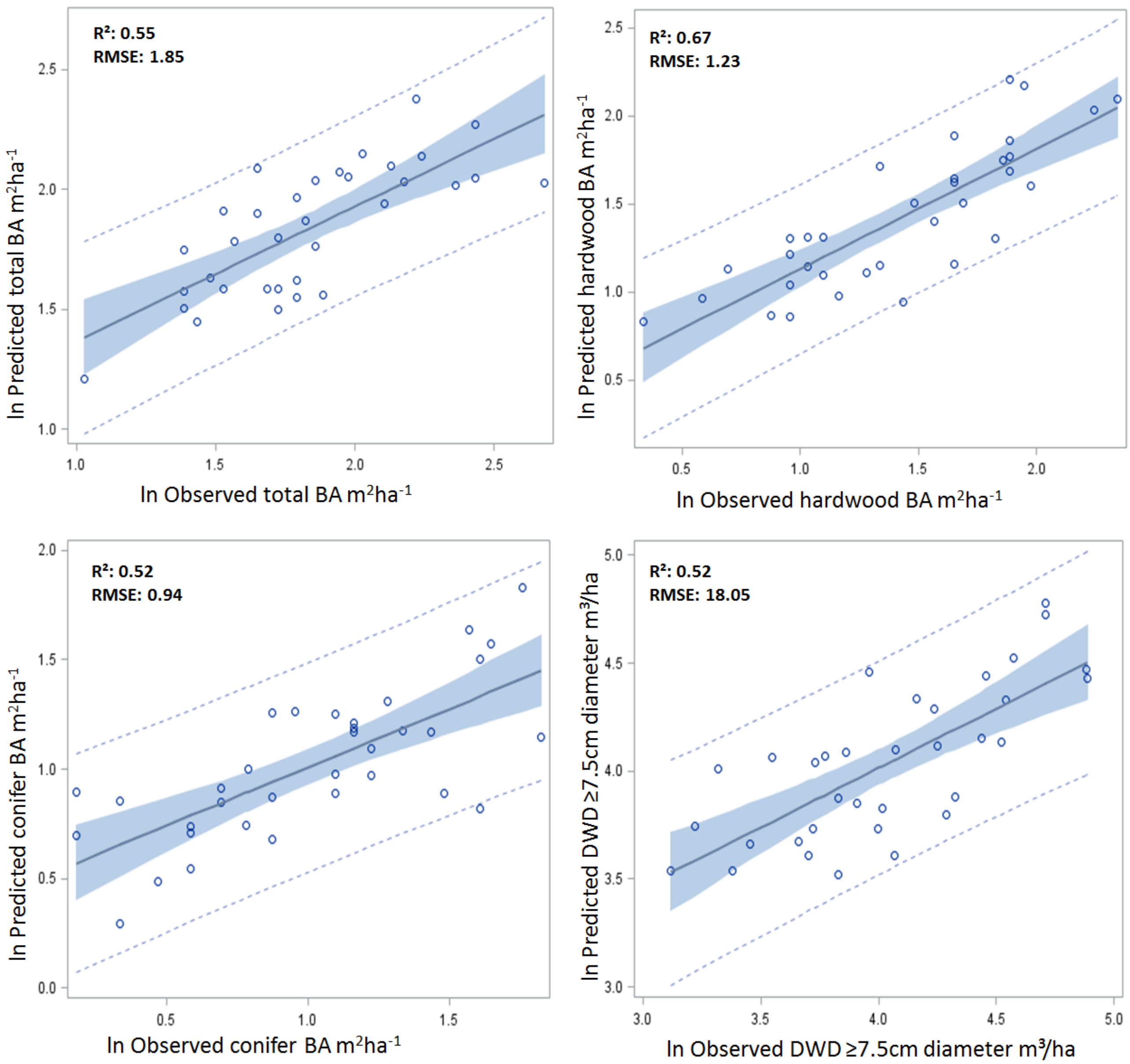

Figure 6. Fit plot of predicted vs. observed total basal area (top left), hardwood basal area (top right), conifer basal area (bottom left), and downed woody debris volume (bottom right).

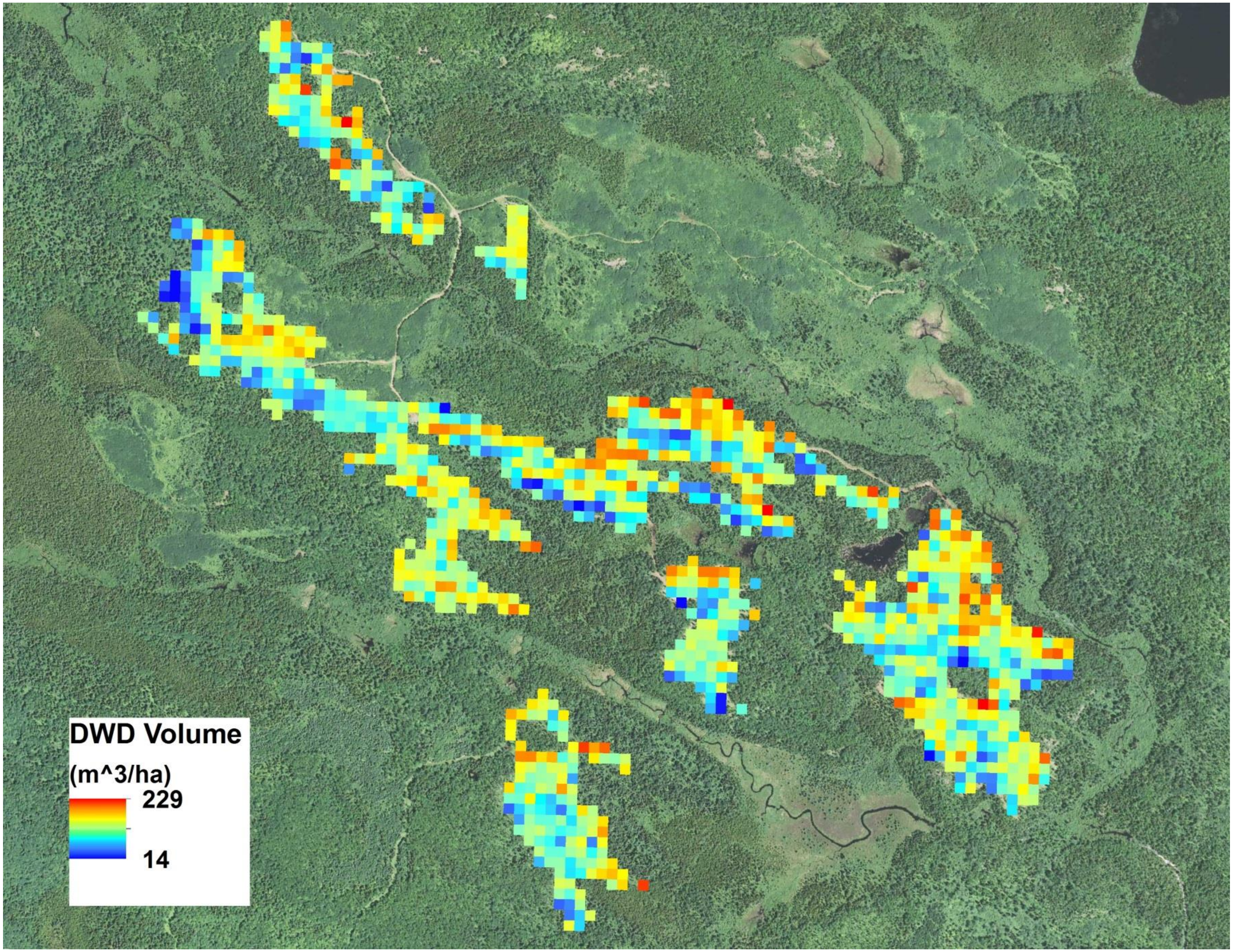

Figure 7. Model results for downed woody debris

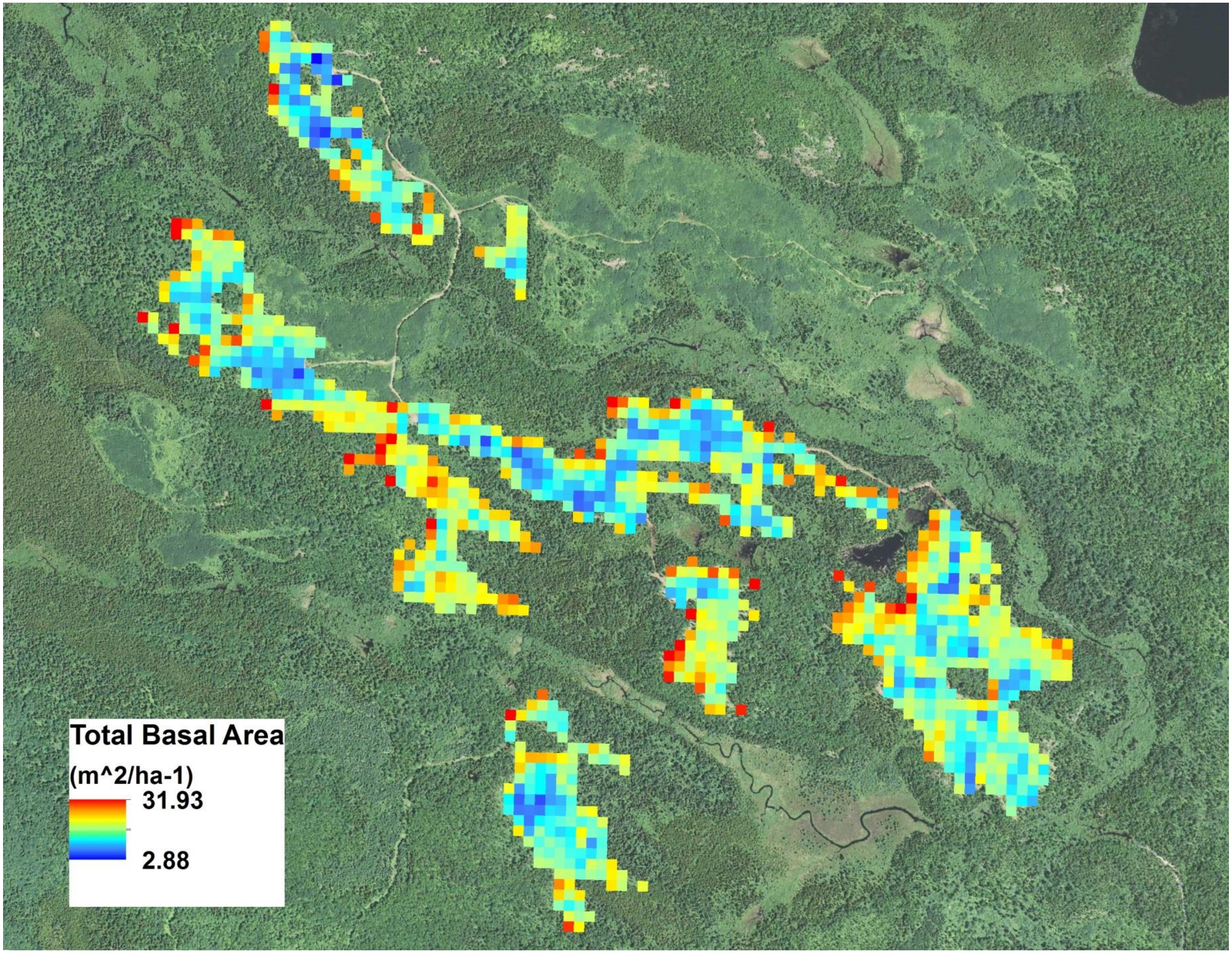

Figure 8. Model results for total basal area

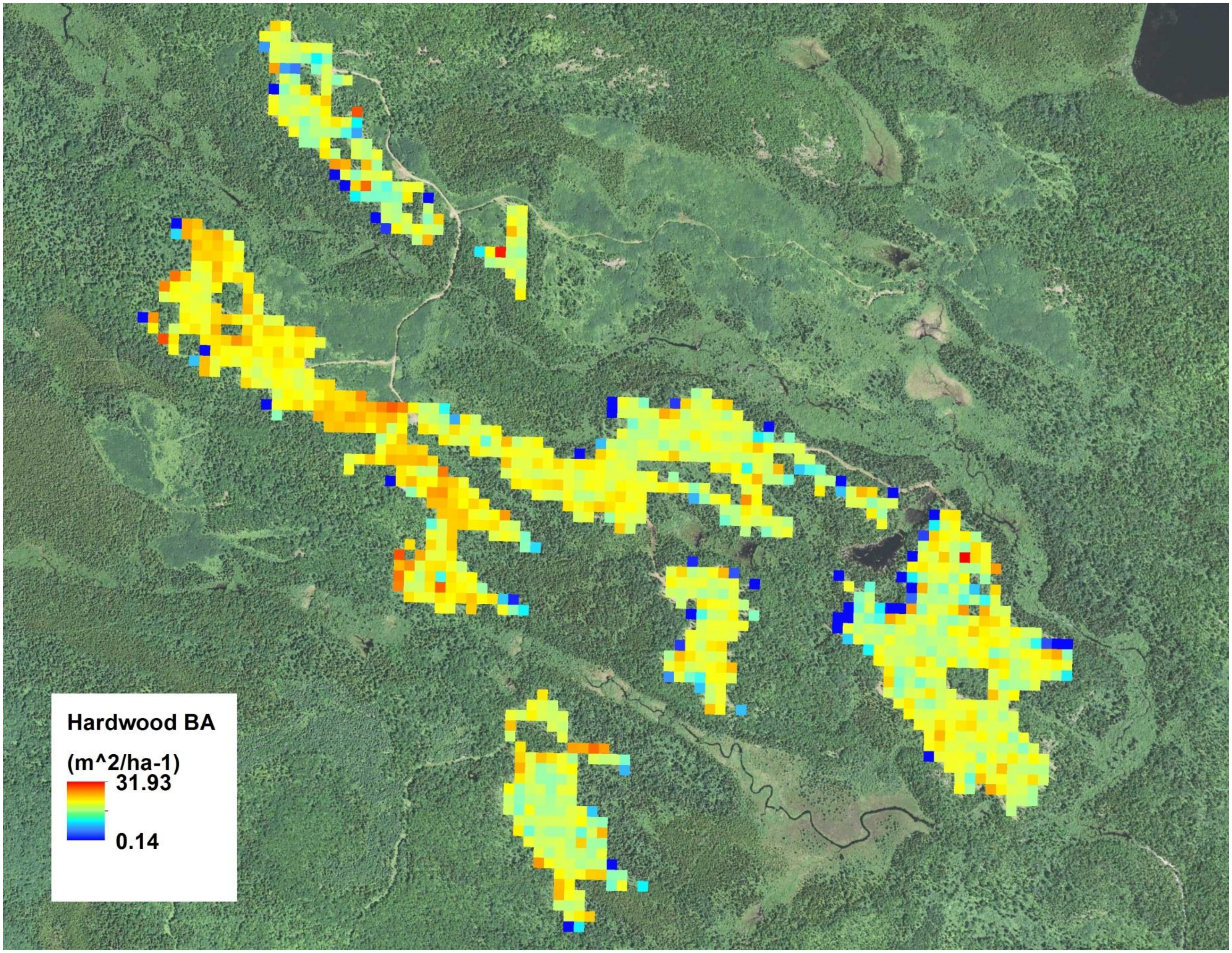

Figure 9. Model results for hardwood basal area.

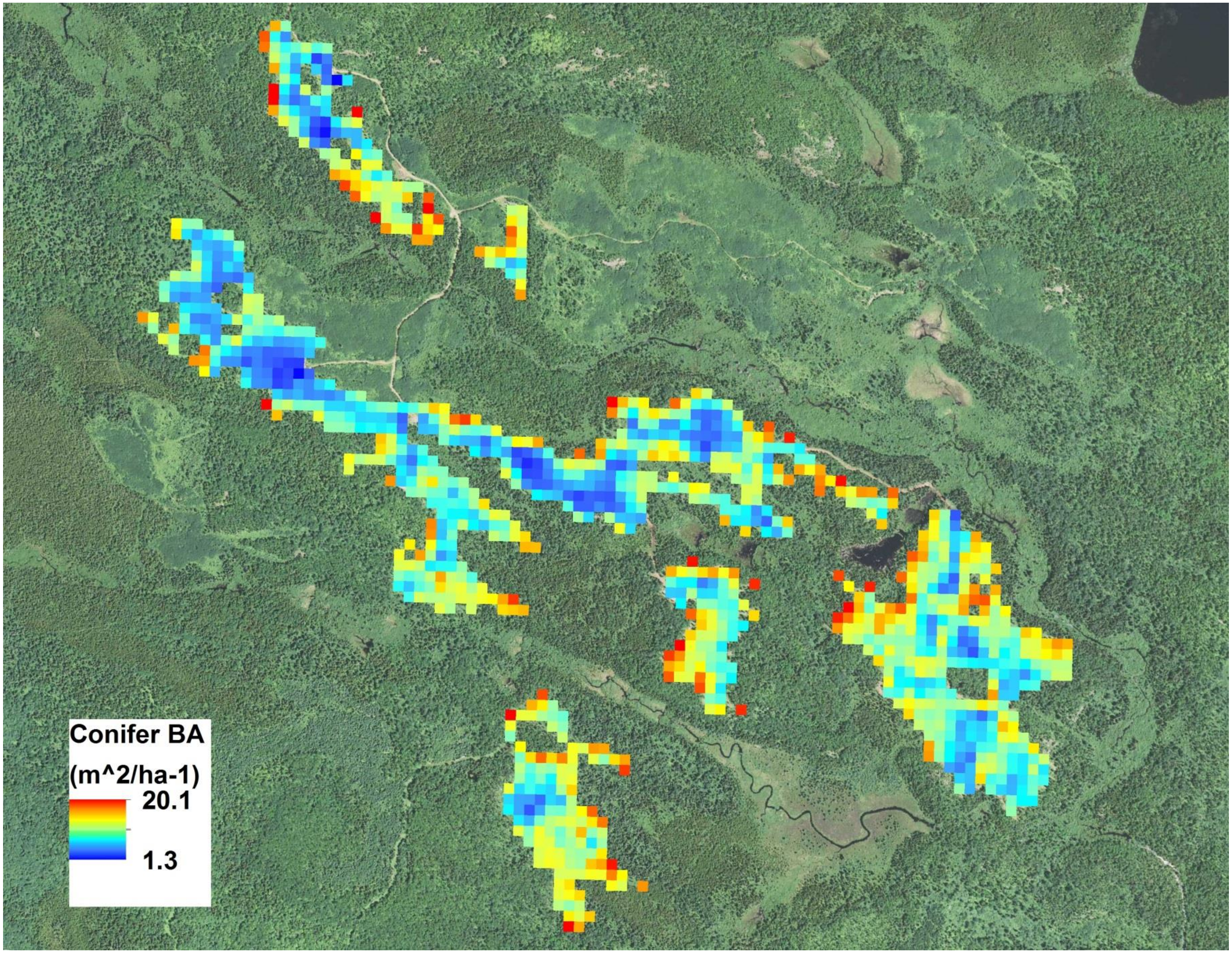

Figure 10. Model results for conifer basal area.

CHAPTER III: GENERAL CONCLUSIONS

Summary

Ecological forestry shows great promise for restoring and maintaining structural and compositional diversity, and increasing resilience and adaptive capacity (D'Amato et al 2011), and balancing ecological, social, and economic values of forests (Lindenmayer et al. 2012); however, there is still a great need for efficient monitoring to better understand the effectiveness of these management strategies and adapt appropriately. This project focused on the use of remote sensing satellite data to quantify and generate models of (1) downed woody debris (DWD) and (2) residual basal area (BA) following retention harvest treatments in northeast Minnesota.

This study found that ground and remote sensing data may be used in combination to calibrate biophysical models of forest structure following retention harvesting, facilitating mapping and extraction of DWD volume and BA in a spatially explicit framework. While not directly comparable, the accuracy of the DWD model calibrations were less than that reported for remote sensing studies using LiDAR to model DWD (van Aardet et al. 2011), and similar to a study using Airborne Synthetic Aperture Radar (AirSAR) (Huang et al. 2009). These results are encouraging and powerful. While ALS technology may have the potential to provide detailed mapping of DWD, this technology is generally expensive, especially if being used to map large areas. With a re-visit time of 16 days and being free and widely available, Landsat imagery is a good option, particularly for large areas. A next step in this research should focus on applying these models to other treatment areas and mature forests with higher standing tree densities, which could allow for landscape-scale modeling of DWD.

While not directly comparable, the accuracy of BA model calibrations were less than that reported for other remote sensing studies in this region focused on mature forest structure (Wolter et al. 2009, Wolter and Townsend 2011, Wolter et al. 2012). This may be partially attributed to the fact that in previous studies using Landsat imagery, field data were collected in forest stands with homogenous species composition and much higher tree density than our field sites. High tree diversity, particularly both hardwood and conifer species at each plot, along with very low density of residual trees likely contributed noise to models. However, our ability to develop these models using Landsat imagery suggests that improved precision are likely possible with the use of more sophisticated sensors such as hyperspectral sensor data acquired by Hyperion (spaceborne NASA sensor) or AVIRIS (Airbroen NASA sensor; Visible/Infrared Imaging spectrometer) and data from high spatial resolution satellite sensor; such as or similar to Quickbird (0.61 – 2.4 m).

Variable retention harvesting has become a widely accepted tool for achieving structural complexity in managed forests and has been applied to forest ecosystems across the globe (Gustafsson et al. 2012). Managing forests for multiple, often conflicting values, in addition to the uncertainty of global climate change may require a flexible approach to maintain ecosystem functioning into the future (White et al. in review). Managing forest ecosystems as complex and adaptive systems may provide this framework (Puettman et al. 2013). A key component of this type of forest management is adaptive management, which allows for shifts in management in response to changing conditions and/or management outcomes (Mladenoff and Pastor 1993); this requires an understanding and knowledge or forest response to management and stressors (Deluca et al. 2010). To successfully manage for complexity in this time of rapid change, an affordable monitoring program that captures key structural elements (e.g. DWD and RBA) is crucial (Cornett and White 2013). The Nature Conservancy has been working on collaborative forest management with public and private owners in the Manitou landscape since 2001. We hope that the ability to produce such remote sensing models will serve as a useful tool for aiding in effective management of structural element within ecological forestry management treatments.

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